# Biological Effects of Agriculturally Derived Surface Water Pollutants on Aquatic Systems—A Review

## C. M. Cooper*

### ABSTRACT

Environmental manipulations and other human activities are major causes of stress on natural ecosystems. Of the many sources of surface water pollutants, agricultural activities have been identified as major contributors to environmental stress, which affects all ecosystem components. In water, agricultural contaminants are most noticeable when they produce immediate, dramatic toxic effects on aquatic life although more subtle, sublethal chronic effects may be just as damaging over long periods. Aquatic systems have the ability to recover from contaminant damage if not seriously overloaded with irreversible pollutants. Thus, contaminant loading level is as important as type of pollutant. Although suspended sediment represents the largest volume of aquatic contaminant, pesticides, nutrients, and organic enrichment are also major stressors of aquatic life. Stream corridor habitat traps and processes contaminants. Loss of buffering habitat, including riparian zones, accelerates effects of pollutants and should be considered when assessing damage to aquatic life. Protection of habitat is the single most effective means of conserving biological diversity. Current available management practices and promising new technology are providing solutions to many contaminant-related problems in aquatic systems.

I N NATURE there is tremendous diversity of both habitats and organisms. In unpolluted natural systems, the distribution of organisms reflects environmental limits — temperature, dissolved oxygen concentrations, light, substrate, nutrients, and other factors. Substrate makes up a major component of aquatic habitat. The diversity of habitats that occurs in nature represents opportunities for adapted organisms to thrive. In diverse natural systems with different habitats and niches, communities of animals and plants have developed and have formed self-balancing systems. These concepts are important when discussing the effects of pollution on aquatic life because pollution in an aquatic system seldom affects a single species of organism. When directly affecting one organism, pollution indirectly affects others by shifting the balance of nature, e.g., changing predator-prey balance. Additionally, pollution can also alter habitat or environmental parameters, thus affecting species or communities and changing balance.

Aquatic systems can be arbitrarily broken into two groups: primary receivers and secondary receivers. The primary receivers include wetlands, small streams, and impoundments that receive runoff and contamination directly from the land. As a result of high concentrations of contaminants from direct inflow, these systems may suffer both acute and chronic effects from pollutants. The secondary receivers include those downstream rivers and larger lakes that receive inflow from multiple tributaries as well as limited direct runoff. Acute effects of pollution in these water bodies are noticeable, but difficult-to-measure chronic problems are more likely to occur.

Agriculture is a major contributor of pollution in the USA. In a recent assessment of lakes cross the country, the USEPA found that 25% of more than 12 million lakes surveyed in 34 states were impaired or partially impaired, and 20% were threatened by pollution mainly from nutrients and sediments (USEPA, 1989). State reports showed that 76% of the pollution entering lakes originated from nonpoint or undescribed sources, 11% came from point sources, and 12% was from natural sources. Although several other sources were major contributors, agriculture was listed as the most frequently cited source. States also reported that agricultural nonpoint sources caused 64% of river water quality degradation, and pesticides have adversely affected almost 5000 water bodies in 609 of the 3137 counties surveyed. Wells (1992) quoted USEPA and USDA saying, "Agricultural stresses, largely from excess nutrients, sediment and pesticides affect 58 percent of impaired lake acres, 55 percent of impaired stream miles and 21 percent of impaired estuarine systems" in the USA.

## STRESSES AND EFFECTS

Although all organisms are adapted to their environments, natural environments periodically stress organ-

USDA-ARS, National Sedimentation Laboratory, P.O. Box 1157, Oxford, MS 38655. Received 19 June 1992. *Corresponding author.

**Abbreviations:** LC, lethal concentration; BOD, biochemical oxygen demand.

isms that inhabit them. Tidal movements create stress twice daily in estuarine and rocky shore habitats. Runoff from natural rainfall can increase stream discharge to the point of causing bed movement. Ephemeral streams undergo an annual cycle of desiccation stress. Aquatic life has adapted to these normally encountered stresses, but is not, in many cases, adapted to additional imposed environmental stresses caused by pollution. Some imposed stresses are similar to natural ones, but many are foreign to the aquatic environment. Consider, for example, low dissolved oxygen. Naturally occurring low concentrations of dissolved oxygen can be found in aquatic habitats such as those draining extensive wetlands with high organic matter. In a low-oxygen environment, fish react by pumping more water over the gill surfaces. When toxic pollutants such as Cu are introduced, an additional stress is created at a critical time. When dissolved oxygen is low and fish pump more water over the gills, they absorb more Cu, which leads to higher metal poisoning during periods of dissolved oxygen stress. Ecosystem complexity results in multiple causality and synergy of multiple stresses acting in the ecosystem simultaneously. For example, researchers have isolated at least 18 induced stressors working in the Green Bay, WI, ecosystem (El-Hinnawi and Hashmi, 1987). Stressors make organisms susceptible to disease and parasites (Wedemeyer et al., 1976). Over a dozen common fish hatchery diseases are stress-mediated (Wedemeyer and Wood, 1974). Many definitions of pollution have been proposed; each hinges on something that changes natural conditions and causes unnatural stress (Hynes, 1960; Holdgate, 1971). Although thermal pollution may mimic water flowing from a hot spring, it is heated water out of place. Edwards (1972) presented a practical definition, saying that pollution is the "release of substances or energy into the environment by man in quantities that damage either his health or his resources."

## AGRICULTURAL INPUTS AND IMPACTS

### Sediments

Instream suspended sediments and bedload are, by volume, the largest pollutants in the USA (Fowler and Heady, 1981). The Mississippi River carries 331 million t of topsoil to the Gulf of Mexico annually (Brown, 1984). In primary receiving streams, suspended material and bedload scour epiphytic communities (organisms that inhabit the sediment surface) and reduce community productivity, sometimes by drastic proportions. Some aquatic communities can adjust to sedimentation; others cannot. In an early study of English lakes, Pearsall (1917, 1918) found that some macrophytes can survive in disturbed areas. Pearsall noted that *Isoetes* was easily smothered by sediment deposition because it apparently could not alter its root level. In many instances, it was replaced by *Potamogeton perfoliatus* during increased silting. In the Weaver Bottoms study, Fremling et al. (1976) documented that increases in sedimentation rates accompanied alternations in flow patterns from dredging in the Mississippi River. This combination destroyed plant and animal communities of the marsh, leaving a barren, windswept body of shallow, unproductive water. In unstable streams of the loess hills of northern Mississippi, increased discharge from large storm events initiates

bedload movement, including sand and gravel, and destroys the periphyton community and benthic invertebrates. Almost no aquatic plant communities form because of excessive sediment loads. Since such movement causes a scour–aggradation cycle, it destroys or buries reproductive habitat for fish, resulting in reproductive failure (Cooper and Knight, 1987).

Benthos and fish are the two most frequently used biotic indicators for aquatic stress. Tebo (1955) showed that high rates of sedimentation reduced benthos, both by mortality and drift. Gammon (1970) documented changes in invertebrate and fish populations in a small stream where sediment load from a crushed limestone quarry was measured. When sediment load increased so that sediments accumulated, benthic populations decreased by 60%. Densities varied from 10 750 organisms/m² above the quarry to 86 organisms/m² below the quarry. Reed (1977) found that siltation from highway construction in Virginia decreased the number of invertebrate and fish species by 23% and the number of organisms by 40%. Cooper conducted a series of evaluations on the effects of suspended sediments on nonmobile invertebrates. He studied an 83 km long stream in an intensely cultivated agricultural watershed and found that bryozoans, especially the environmentally sensitive *Pectinatella magnifica*, progressively disappeared in downstream areas with degraded habitat and increased total solids (Cooper and Burris, 1984). Cooper also documented that sediments were detrimental to stream benthos by eliminating sediment-sensitive organisms during periods of sediment deposition, especially the more sensitive larval stages. Benthic productivity was limited mainly by lack of suitable substrate because of accumulation of fine sediments (Cooper, 1987). He then tested mortality of three species of sessile invertebrates and found that the invertebrates could withstand high concentrations of suspended sediments (>1000 mg/L) for brief periods but that mortality rates increased rapidly with time (Fig. 1 and 2). The 96 h $LC_{50}$ (lethal concentration to 50% of the population) was 200 to 750 mg/L for the sponges and bryozoans (Cooper, 1988).

In lakes and reservoirs, suspended sediments can limit primary productivity of algae. While some "low light" phytoplankton require "shaded" conditions (Tilzer et al., 1976), the growth of most limnetic algae are dampened by light reduction from suspended sediments. Depth of light penetration into the water column is also reduced by suspended sediments, shrinking the photic zone. In a long-term water quality study of Lake Chicot, AR, Cooper et al. (1991) compared a large (south) lake receiving drainage from a 930 km² agricultural watershed with an isolated (north) third of the lake, which received only ephemeral inflow from a predominantly agricultural watershed of <100 km². Mean suspended sediments for the flow-through south lake were 208 mg/L for the period 1980 to 1985. For the isolated north lake, they were 47 mg/L. While chlorophyll cycles represented the results of many variables, phytoplankton production was limited seasonally in the main basin by suspended sediments. In a typical year light was limited by suspended sediments in the south lake so that phytoplankton production was dampened despite excessive nutrients. As sediments were removed from the water column in mid-June and early July, light penetration increased (Stefan et al., 1983) and
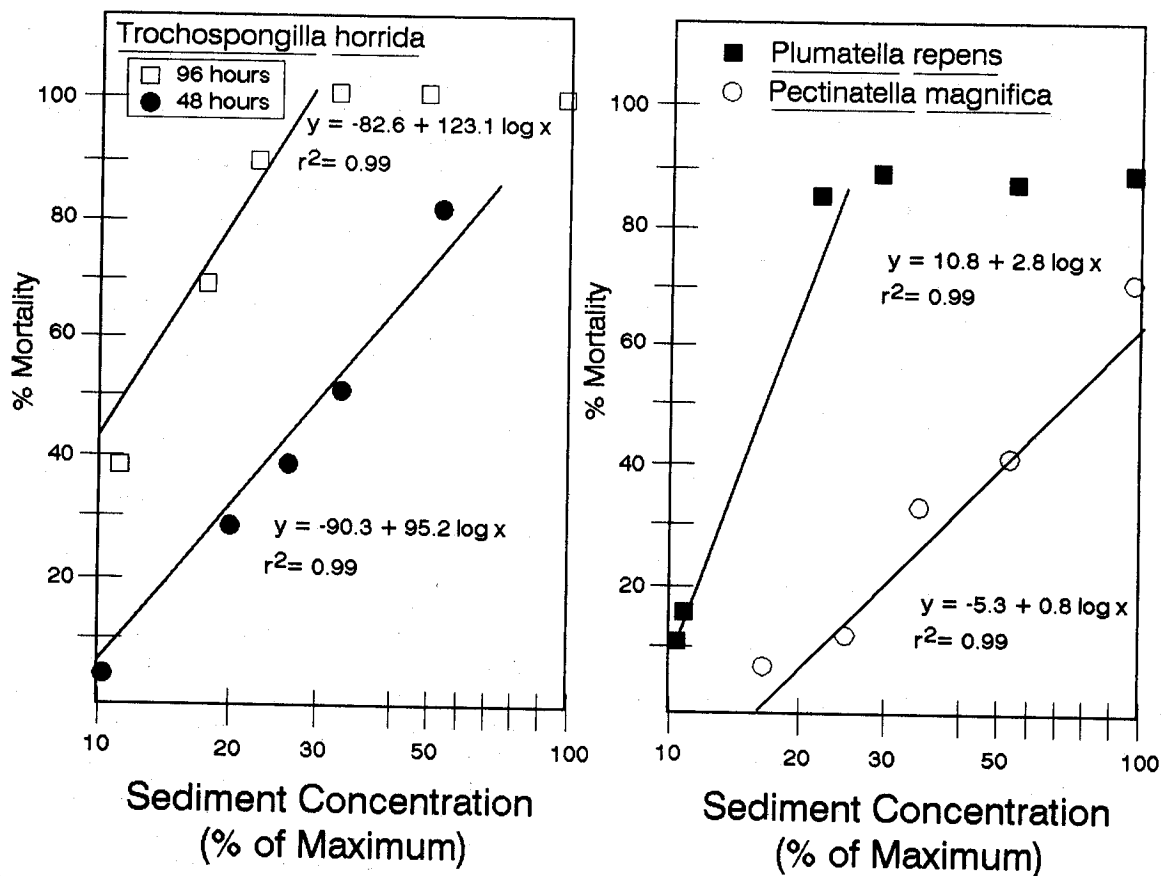
**Fig. 1.** Concentration mortality plots for the poriferan, *Trochospongilla horrida*, for 48 and 96 h exposure to suspended sediments (100% = 5017 mg/L) (after Cooper, 1988).
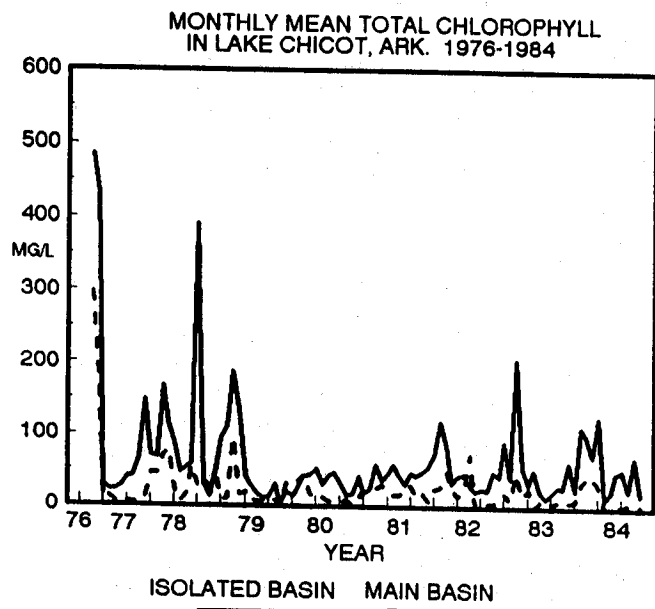


**Fig. 2.** Concentration mortality plot for the bryozoans *Plumatella repens* and *Pectinatella magnifica* for 96 h exposure to suspended sediment (100% = 1000 mg/L) (after Cooper, 1988).
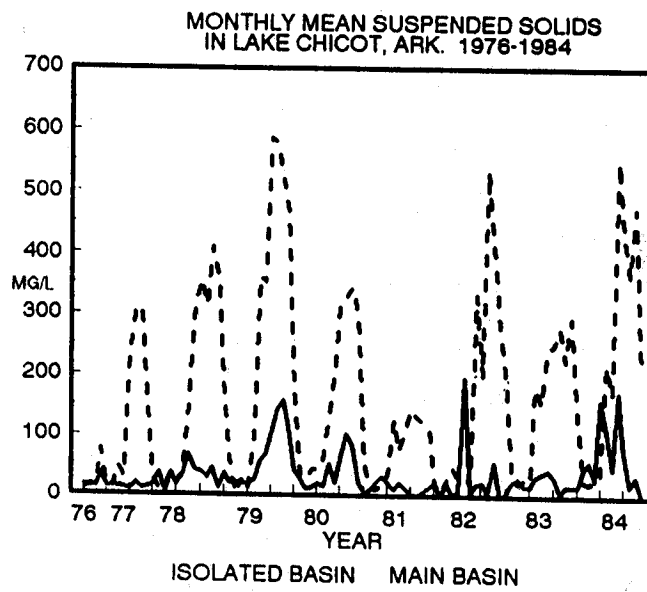


**Fig. 3.** Chlorophyll and sediment concentrations for the main body and the isolated north basin from 1976 to 1984.

chlorophyll concentrations also began to increase (Fig. 3). In the north lake where sediments were not normally a problem, chlorophyll peaks were generally associated with spring and fall turnover or summer temperature maxima, and peaks were stimulated by storm events. North basin fisheries were outstanding. Conversely, primary productivity in the main basin seldom approached potential, and the fish community was composed mainly of less desirable species of scavengers.

Sediment deposition can have a major detrimental effect by degradation or total loss of habitat. Reelfoot Lake, Tennessee, contained 20 250 ha when it was formed in 1811 by an earthquake. It now has only 7300 ha and may be filled in 50 yr (Batie, 1984). While successional filling is the fate of all lakes, natural lake life is generally measured in thousands of years.

## NUTRIENTS

Nutrients from agriculture represent a second contaminant group that affects aquatic life. Crop lands are major contributors since a requirement for satisfactory crop production is nutrient availability, which is normally acquired by commercial fertilizer application. Runoff from confined feeding operations is an additional rural contributor (Loeher, 1984). In urban areas, excessive fertilization of lawns and sewage discharge can be significant contributors of nutrients (Weibel, 1969).

Plant growth in water is limited by some physical or chemical factor. Sunlight provides energy for aquatic plants. Availability of P and N generally limits growth in fresh waters, as it does in crop production (Vollenweider, 1968). Conversely, excessive P and N contribute to growth of plants in water. Eutrophication, the result of excessive nutrients, is not necessarily good or bad. In proper context, it is a natural phenomenon. However, accelerated cultural eutrophication that does not allow for functional water use and shortens lake life generally can be traced directly to human activity (Hutchinson, 1957).

Excessive plant growth from eutrophication creates several problems. The blanketing effects of macrophytes and algae can alter faunal species composition because of physical changes in habitat. Some algal species associated with eutrophication produce toxins. Respiration of plants depresses dissolved oxygen at night and even during daylight if light penetration is reduced. Decay of plant biomass can have the same effect as large inputs of organic matter, further reducing oxygen. Thus, excessive plant growth can deplete oxygen concentrations below the 4 mg/L limit for warmwater fishes and result in fish kills (Wedemeyer et al., 1976). Oxygen depletion is probably the greatest source of stress associated with eutrophication. Short-term deficits may be readily observable as with fish kills, but long-term reductions may have even more drastic results. These latter deficiencies create shifts in benthic communities and alter the primary consumer level of aquatic life (Cooper, 1980). Some nutrients are directly toxic to aquatic life. Ammonia ($NH_3$) from livestock waste may be toxic to fishes at levels as low as 0.02 mg/L, especially at high pH values.

Nutrient concentrations in streams are closely related to land use (Omernik, 1977), but streams, as primary receivers, are influenced many times by overriding physical factors. Stream characteristics such as slope, depth, current velocity, canopy, and stream order may determine phytoplankton density regardless of nutrient availability.

## ORGANIC CONTAMINATION

Excessive organic wastes are among the oldest and most widespread forms of water pollution. Organic wastes have a high proportion of solids, which can rapidly blanket benthic habitat. The development of profuse growths of "sewage fungus," a community of heterotrophic microorganisms, is a characteristic indication of organic contamination (Hellawell, 1986). The most noticeable consequence of organic contamination is its effect on dissolved oxygen concentrations in water and sediments. Large amounts of organic matter from animal wastes provide a greater BOD (biochemical oxygen demand) than an aquatic system can supply oxygen for, resulting in an "oxygen-sag." Nutrient loading is frequently associated with organic enrichment, resulting in a compounded problem. Natural waters commonly have a BOD of 0.5 to 7 mg/L (Klein, 1959), whereas chicken wastes may have a BOD of 24 000 to 67 000 mg/L (Weller and Willetts, 1977). When organic matter exceeds the capacity of a system to assimilate it, a degradation cycle begins. Initially, the enhanced level of organic material stimulates increased activity of aerobic decomposers. When the rate of oxygen consumption by aerobic decomposers exceeds the rate of reaeration, dissolved oxygen concentration begins to fall, and some species are eliminated. If the dissolved oxygen decline continues, aerobic decomposers cease to function and anaerobic organisms populate the sediment and the water.

The significance of low dissolved oxygen reaches beyond changes in species composition because oxygen concentration affects chemical reactions. Ammonia provides an example. Under aerobic conditions, nitrifying bacteria are dominant, and ammonia is converted to nitrite by bacteria (*Nitrosomonas* sp.) and then to nitrates by *Nitrobacter* sp. In this cycle, toxic ammonia is oxidized to readily available nutrients. Under anaerobic conditions denitrification usually occurs by other bacteria such as *Thiobacillus denitrificans* (Abel, 1989).

## PESTICIDES AND METALS

Pesticides have played a major role in the world's struggle against food shortages and vector-borne diseases. Our society and economy would be radically different without them. However, pesticides present a continuing problem of contamination. The lethal effects of pesticides have been well documented since toxicity testing is necessary before a pesticide can be registered for general use. In many cases documentation of persistence or biotic uptake of currently used pesticides has been limited to initial preregistration testing and has not included impact from repeated use. With long-term use, pesticides are more likely to be present in low but consistent levels (Cooper, 1991) and may pose unknown, sublethal problems. Approximately 200 million kilograms of pesticides were applied to the 10 major field crops in the USA in 1988 (Anonymous, 1989). Often less than 0.1% of pesticides applied to crops actually reach the target organisms (Pimental and Levitan, 1986). That leaves more than 99% of applied material to be degraded or potentially contaminate air, soil, or water. While actual contamination is lower, many pesticides can be traced from application sites. Ten years after application, organochlorine insecticides remained detectable in soils, wetlands, lake water and sediments, and fish in Moon Lake watershed in Mississippi (Table 1). Residual insecticide concentrations were significantly higher in aqueous–sediment phases of surface waters during the winter–spring wet season than during dry season. Significantly increased concentrations in runoff suggest the magnitude of the DDT source that remains in watershed

Table 1. Concentrations of DDT and toxaphene† residues in major watershed components in Moon Lake, Mississippi, and its watershed (after Cooper and Lipe, 1992).

| Component | DDT | Toxaphene |
|---|---|---|
| | μg/kg | |
| Soil $n=69$ | 369.31 ( 0 – 6406.93) SD ± 981.19 | 734.01 ( 0 – 7377.72) SD ± 1099.22 |
| Wetland $n=20$ | 212.84 ( 0 – 568.14) SD ± 176.07 | Not detected |
| Sediment $n=45$ | 235.45 ( 0 – 650.80) SD ± 158.90 | 12.42 (2.6 – 35.80) SD ± 11.56 |
| Water $n=255$ | 0.11 ( 0 – 1.0) SD ± 0.11 | 0.01 ( 0 – 0.40) SD ± 0.06 |
| Fish‡ $n=28$ | 243.96 (80 – 463.0) SD ± 114.03 | Not determined |

† Toxaphene = chlorinated camphene containing 67 – 69% chlorine.
‡ Mississippi Department of Wildlife, Fisheries and Parks.

soil (mean = 369 μg/kg) and emphasizes the importance of watershed management on long-term ecosystem quality. Also in Moon Lake, pesticides in current use that rapidly degrade were found in water, sediments, and fish (Cooper, 1991). Permethrin [m-phenoxybenzyl cis, trans-( ± )-3-(2,2-dichlorovinyl)-2,2-dimethylcyclopropane-carboxylate], fenvalerate [alpha cyano-3-phenoxybenzyl 2-(4-chlorophenyl)-3-methylbutyrate], and methyl parathion [O,O-dimethyl O-(p-nitrophenyl) phosphorothioate] were found in water, wetland, and lake sediments, and fish during and after spray season. Anderson (1982) documented behavioral changes or death in freshwater invertebrates at surface water concentrations as low as 0.022 μg/L for fenvalerate and 0.03 μg/L for permethrin. He found that the range of accumulated fenvalerate in snails was 177 to 1286 times greater than water concentration. Some insecticides in current use not only accumulate, they can be as toxic as and often more toxic than banned organochlorines. Parathion and malathion have approximately the same acute toxicity ($LC_{50}$) as DDT [1,1,1-trichloro-2,2-bis(p-chlorophenyl)-ethane] to cladoceran Crustacea in the genera Daphnia and Simocephalas (Sanders and Cope, 1966).

Herbicides have received less environmental research attention than insecticides because of their lower acute toxicity to animals. Although off-site problems from herbicides have rarely been documented in aquatic systems, residues of several frequently used herbicides are common in agricultural drainages. Baker and Richards (1989) found atrazine [2-chloro-4-(ethylamino)-6-(isopropylamino)-s-triazine], alachlor [2-chloro-2',6'-diethyl-N-(methoxymethyl)acetanilide], and metolachlor [2-chloro-N-(2-ethyl-6-methylphenyl)-N-(2-methoxy-1-methylethyl) acetamide] routinely in Ohio streams during a 4-yr study of a corn (Zea mays L.) and soybean [Glycine max (L.) Merr.] producing area, but they found no incidences of adverse effects from herbicides on aquatic animal communities or human health. Long-term exposure to atrazine at 0.7 mg/L for several months produced no mortality among bluegills (Lepomis macrochirus) (Macek et al., 1976). However, bluegills began to lose their equilibrium at 0.2 mg/L, spawning was reduced in fathead minnows (Pimephales promelas), and brook trout (Salvelinus fontinalis) fry grew more slowly than normal.

Chronic toxicity effects on aquatic communities from low levels of herbicides are difficult to detect. Some herbicides reduce primary productivity by being toxic to phytoplankton. Others produce indirect effects when extensive organic matter from dead plants uses available oxygen. When aquatic herbicides are used to destroy aquatic macrophyte communities, the change in habitat results in major alterations in community structure.

The first modern pesticides were metals, specifically As and Hg. The earliest widespread arsenical insecticide was paris green, a vibrant pigment that was 40% As (McEwen and Stephenson, 1979). It was first used against the Colorado potato beetle (Lepinotarsa decemlineata) in 1865. Since it was highly toxic to animals and plants, it was replaced by lead arsenate before 1900. Mercury was historically used as a seed treatment and fungicide. Unlike synthetic pesticides, metals do not degrade; they may oxidize or be bonded in chemical reactions, but they do not disappear. Mercury was used until 1985, and As is still applied in limited quantities. Both are found in low concentrations as naturally occurring elements, but increased levels in stream and lake sediments in agricultural regions are cause for environmental concern.

## SOLUTIONS

Agricultural water quality priorities must include solutions to problems of surface water pollution inputs and resolution of downstream problems. Solutions to surface water problems caused by agriculture center around best management practices and sound land use policy. Land use policy should dictate that land be used only within the capacity to support agricultural production without abuse. Sediment-related contamination is best prevented by a series of practices that either prevent erosion or trap sediment. These include cover crops, various degrees of conservation tillage, grassed waterways, terraces, filter strips, riparian zones, contour farming, and water and sediment control basins. Mutchler and McDowell (1990) evaluated winter cover crops for cotton (Gossypium hirsutum L.) land. They found that annual soil loss was reduced from 74 t/ha·yr to 20 t/ha·yr and runoff was reduced from 48 to 26% of total runoff when winter cover crops were established. Dendy and Cooper (1984) measured a 77% sediment trapping efficiency for a 1.09 ha farm pond, which drained a mixed pasture–row crop watershed.

Aquatic habitat is also degraded by sediments from channel banks and beds in unstable streams. Grissinger et al. (1990) suggested that better than 80% of the total sediment yield for Goodwin Creek in northern Mississippi originates as channel and gully erosion. Construction efforts used to stabilize stream channels can also be used to replace lost or degraded habitat. Cooper and Knight (1987) found that grade control structures in northern Mississippi provided stable, quality habitat that resulted in improved stream ecosystems in unstable streams. The structures added vital reproductive habitat and a supporting community of food organisms. Knight and Cooper (1990) evaluated different rock placement configurations used for stream training and stabilization. They reported that transverse stone dikes or groins enhanced habitat in unstable streams so that treated stream reaches were comparable to natural reaches in both numbers and weight of fish. Scour holes associated with transverse dikes did not affect fish production directly,

but provided additional areas capable of supporting more fish and larger fish as well as associated food communities.

Duttweiler and Nicholson (1983) estimated that 1.16 million t of P and 4.65 million t of N are discharged with runoff and sediment from North American croplands. Many of the management practices that reduce sediment loads help reduce nutrients. Proper application following soil testing can reduce potential N discharge by 35 to 94% (Anonymous, 1979). Farm ponds and impoundments can trap more than 70% of the nutrients that flow into them (Cooper and Knight, 1990) from agriculture or pasture. Schofield et al. (1990) evaluated dairy farming practices to identify those having the greatest impact on water quality. They documented that dairy yard and parlor washing caused the most significant deterioration of downstream water quality. Constructed wetlands are a rapidly spreading technology that can filter and process waste from confined livestock operations like milking parlor washoff. Cooper (1992, unpublished) found that three constructed wetland cells removed 91% of ammonia, 62% of total organic P, and 76% of BOD when coupled behind an anaerobic lagoon during their first season of operation at a dairy farm.

Instream and reservoir techniques for reducing nutrient impacts are limited because of economic realities. Fortunately, natural processes such as uptake by plankton and adsorption by sediments reduce surface water nutrient levels, and nutrient impacts are lessened when supplies are reduced. Krumholz and Minckley (1984) recorded marked improvement in water quality and fish population in the upper Ohio River only 11 d after the closure of steel industries. Apparently, clean water fish moved into the river from nearby unpolluted habitat as rapidly as water quality improved. Riparian zones reduce surface nutrient inputs and also use nutrients from the shallow groundwater that contributes lateral flow streams (Lowrance et al., 1984). In an ongoing USDA study of a river system in Mississippi, channelized tributaries with little or no riparian buffer zones had two to three times the annual nitrate concentration of natural stream reaches with wetland or riparian buffers in 1989 to 1991 (Cooper, 1991, unpublished).

Organic wastes and pesticides as pollutants are much like nutrients in that once they enter aquatic waters, little can be done to lessen their effects. Organic wastes must be degraded on site by such management practices as constructed wetlands and soil incorporation for enrichment purposes. Although some pesticides degrade rapidly, metals used in pesticide formulations will remain in the environment for the foreseeable future. This forces us, as caretakers of our natural resources, to use agricultural chemicals with care, observing all cautions and to continue to search for innovative approaches for use in application and degradation.

In evaluating common agricultural contaminants and their effects on aquatic systems, alteration or destruction of habitat and results of toxic contamination are the two largest problems. Likewise, these are the two areas where we can make the most progress in restoring aquatic systems or preventing future contamination. To successfully protect downstream water resources including their biota, our objectives must encompass watershed management in addition to stream and lake oversight.

## REFERENCES

Abel, P.D. 1989. Water pollution biology. John Wiley & Sons, New York.

Anderson, R.L. 1982. Toxicity of fenvalerate and permethrin to several nontarget aquatic invertebrates. Environ. Entomol. 11:1251–1257.

Anonymous. 1979. Water quality and agriculture — a management plan. Carolina Soil and Water Conserv. Commission and 208 Agric. Task Force, Raleigh, NC.

Anonymous. 1989. Pesticides. USDA, Econ. Res. Serv. Situation and Outlook Rep. AR-13. USDA, Washington, DC.

Baker, D.B., and R.P. Richards. 1989. Herbicide concentration patterns in rivers draining intensively cultivated farmlands of north western Ohio. p. 103–120. In Pesticides in terrestrial and aquatic environments. Proc. of a National Research Conf., Blacksburg, VA. May 1989. Virginia Water Resour. Res. Ctr., Blacksburg, VA.

Batie, S.S. 1984. Soil erosion: Crisis in America's croplands. The Conservation Foundation, Washington, DC.

Brown, L.R. 1984. Global loss of topsoil. J. Soil Water Conserv. 39:162–165.

Cooper, C.M. 1980. Effects of abnormal thermal stratification on a reservoir benthic macroinvertebrate community. Am. Midl. Nat. 103:149–154.

Cooper, C.M. 1987. Benthos in Bear Creek, Mississippi: Effects of habitat variation and agricultural sediments. J. Freshwater Ecol. 4:101–113.

Cooper, C.M. 1988. The toxicity of suspended sediments on selected freshwater invertebrates. Verh. Int. Verein. Limnol. 23:1619–1625.

Cooper, C.M. 1991. Insecticide concentrations in ecosystem components of an intensively cultivated watershed in Mississippi. J. Freshwater Ecol. 6:237–247.

Cooper, C.M., and J.W. Burris. 1984. Bryozoans — possible indicators of environmental quality in Bear Creek, Mississippi. J. Environ. Qual. 13:127–130.

Cooper, C.M., and S.S. Knight. 1987. Fisheries in man-made pools below grade-control structures and in naturally occurring scour holes of unstable streams. J. Soil Water Conserv. 42:370–373.

Cooper, C.M., and S.S. Knight. 1990. Nutrient trapping efficiency of a small sediment detention reservoir. Agric. Water Manage. 18:149–158.

Cooper, C.M., and W.M. Lipe. 1992. Water quality and agriculture: Management examples from Mississippi. J. Soil Water Cons. 47:220–223.

Cooper, C.M., F.R. Schiebe, and J.C. Ritchie. 1991. Case study of a sediment limited lake. p. 13–26–13–33. In S.-S. Fan and Y.-H. Kuo (ed.) Proc. 5th Federal Interagency Sedimentation Conf., Las Vegas, NV. 18–21 Mar. 1991. Fed. Energy Reg. Comm., Washington, DC.

Dendy, F.E., and C.M. Cooper. 1984. Sediment trap efficiency of a small reservoir. J. Soil Water Conserv. 39:278–280.

Duttweiler, D.W., and H.P. Nicholson. 1983. Environmental problems and issues of agricultural non-point source pollution. p. 13–16. In F.W. Schaller and G.W. Bailey (ed.) Agricultural management and water quality. Iowa State Univ. Press, Ames, IA.

Edwards, R.W. 1972. Pollution. Oxford Biology Readers No. 31. Oxford Univ. Press, Oxford, UK.

El-Hinnawi, E., and M.H. Hashmi. 1987. The state of the environment. Butterworth, London.

Fowler, J.M., and E.O. Heady. 1981. Suspended sediment production potential on undisturbed forest land. J. Soil Water Conserv. 36:47–49.

Fremling, C.R., D.N. Nielsen, D.R. McConville, and R.N. Vose. 1976. The Weaver Bottoms: A field model for the rehabilitation of backwater areas of the upper Mississippi River by modification of standard channel maintenance practices. Final Rep. to U.S. Army Corps of Engineers, St. Paul District, St. Paul, MN.

Contracts DACW37–75–C–0193 and –0194. Biology Dep., Winona State Univ., Winona, MN.

Gammon, J.R. 1970. The effect of inorganic sediment on stream biota. Rep. W72–00851 (EPA/18050- DWC-12/70). DePauw Univ., Greencastle, IN.

Grissinger, E.H., A.J. Bowie, and J.B. Murphey. 1990. Goodwin Creek bank instability and sediment yield. p. PS-32–PS-37. *In* S.-S. Fan and Y.-H. Kuo (ed.) Proc. 5th Federal Interagency Sedimentation Conf., Las Vegas, NV. 18–21 Mar. 1991. Fed. Energy Reg. Comm., Washington, DC.

Hellawell, J.M. 1986. Biological indicators of freshwater pollution and environmental management. Elsevier, London.

Holdgate, M.W. 1971. The need for environmental monitoring. p. 1–8. *In* Int. Symp. on Identification and Measurement of Environmental Pollutants, Ottawa, Ontario, Canada. June 1971.

Hutchinson, G.E. 1957. A treatise on limnology. Vol. 1. John Wiley & Sons, New York.

Hynes, H.B.N. 1960. The biology of polluted waters. Liverpool Univ. Press, Liverpool, England.

Klein, L. 1959. River pollution. Vol 1. Butterworth, London.

Knight, S.S, and C.M. Cooper. 1990. Effects of bank protection on stream fishes. p. 13–34–13–39. *In* S.-S. Fan and Y.-H. Kuo (ed.) Proc. 5th Federal Interagency Sedimentation Conf., Las Vegas, NV. 18–21 Mar. 1991. Fed. Energy Reg. Comm., Washington, DC.

Krumholz, L.A., and W.L. Minckley. 1984. Changes in the fish population in the upper Ohio River following temporary pollution abatement. Trans. Am. Fish. Soc. 93:1–5.

Loeher, R.C. 1984. Pollution control for agriculture. Academic Press, Orlando, FL.

Lowrance, R., R.L. Todd, J. Fail, Jr., O. Hendrickson, Jr., R. Leonard, and L. Asmussen. 1964. Riparian forest as nutrient filters in agricultural watersheds. BioScience 34:374–377.

Macek, K.J., K.S. Buxton, S. Sauter, S. Gnilka, and J.W. Dean. 1976. Chronic toxicity of atrazine to selected aquatic invertebrates and fishes. USEPA Rep. 600/3–76–047. USEPA, Washington, DC.

McEwen, F.L., and G.R. Stephenson. 1979. The use and significance of pesticides in the environment. John Wiley & Sons, New York.

Mutchler, C.K., and L.L. McDowell. 1990. Soil loss from cotton with winter cover crops. Trans. ASAE 33:432–436.

Omernik, J.M. 1977. Nonpoint source–stream nutrient level relationships: A nationwide study. USEPA Rep. 600/3–77–105. USEPA, Natl Env. Res. Lab., Corvallis, OR.

Pearsall, W.H. 1917. The aquatic and marsh vegetation of Esthwaite water. J. Ecol. 5:180–202.

Pearsall, W.H. 1918. The aquatic and marsh vegetation of Esthwaite water: II. J. Ecol. 6:53–74.

Pimental, D., and L. Levitan. 1986. Pesticides: Amount applied and amount reaching pests. BioScience 36:38.

Reed, J.R., Jr. 1977. Stream community response to road construction sediments. Virginia Water Resour. Res. Ctr. Bull. 97.

Sanders, H.O., and O.B. Cope. 1966. Toxicities of several pesticides to two species of cladocerans. Trans. Am. Fish. Soc. 95:165–169.

Schofield, J.S., and R.P. Merriman. 1990. Impact of intensive dairy farming activities on river quality: The eastern Cleddan catchment study. J. Inst. Water Environ. Manage. 4:176.

Stefan, H.G., J.J. Cardoni, F.R. Schiebe, and C.M. Cooper. 1983. Model of light penetration in a turbid lake. Water Resour. Res. 19:109–120.

Tebo, L.B. 1955. Effects of siltation resulting from improper logging, on the bottom fauna of a small trout stream in the southern Appalachians. Prog. Fish Cult. 17:64–70.

Tilzer, M.M., C.R. Goldman, R.C. Richards, and R.C. Wrigley. 1976. Influence of sediment inflow on phytoplankton primary productivity in Lake Tahoe (California-Nevada). Int. Rev. Gesamten Hydrobiol. 61:169–181.

U.S. Environmental Protection Agency. 1989. Report to Congress; Water Quality of the Nation's Lakes. USEPA Rep. 440/5–89–003. Office of Non-point Source Control Branch, Washington, DC.

Vollenweider, R.A. 1968. Scientific fundamentals of the eutrophication of lakes and flowing waters, with particular references to phosphorus and nitrogen as factors in eutrophication. Organization for Econ. Coop. and Development, Paris.

Wedemeyer, G.A., F.P. Meyer, and L. Smith. 1976. Environmental stress and fish diseases. T.F.H. Publ., Neptune City, NJ.

Wedemeyer, G.A, and J.W. Wood. 1974. Stress as a predisposing factor in fish diseases. Fish. Leafl. 38. U.S. Fish Wildl. Serv.

Weibel, S.R. 1969. Urban drainage as a factor in eutrophication. p. 383–403. *In* Eutrophication: Causes, consequences, corrections. National Academy of Sciences, Washington, DC.

Weller, J.B., and S.L. Willetts. 1977. Farm wastes management. Crosby Lockwood Staples, London.

Wells, H.W. 1992. Pollution prevention. Pollut. Eng. 24:23–25.